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Linking Biophysical and Economic Assessments of Ecosystem Services for a Social–Ecological Approach to Conservation Planning: Application in a Biosphere Reserve (Biscay, Spain)

Nekane Castillo-Eguskitza ^{1,*}, María F. Schmitz ², Miren Onaindia ¹ and Alejandro J. Rescia ²

¹ Plant Biology and Ecology Department, University of the Basque Country UPV/EHU, 48940 Biscay, Spain; miren.onaindia@ehu.eus

² Department of Biodiversity, Ecology and Evolution, Complutense University of Madrid, 28040 Madrid, Spain; ma296@bio.ucm.es (M.F.S.); alejo296@bio.ucm.es (A.J.R.)

* Correspondence: nekane.castillo@ehu.eus

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Abstract: The search for a balance between nature conservation and sustainable development remains a scientific and spatial planning challenge. In social-ecological systems based on traditional rural activities and associated with protected areas, this balance is particularly complex. Quantifying the economic impact of land use changes on ecosystem services can be useful to advise policy makers and improving social-ecological sustainability. In this study, we evaluated the land use changes in a time series and estimated the monetary value of the ecosystem services of the Urdaibai Biosphere Reserve (Biscay, Spain). In addition, we linked the monetary and biophysical values of land uses in each zoning units of the reserve, in order to identify the spatial adjustment between both assessments. Results showed that land use changes have clearly homogenized the landscape without substantially affecting its economic value. The methodological approach allowed detection that the reserve zoning was performed based more on its biophysical values than on economic ones. Thus, evident divergences between the biophysical and economic assessments were found. The core area was the one that had the highest coincidences (medium values) between both ecosystem services assessments, which highlights its importance not only in biophysical terms, is also economical. The procedure followed proved to be a useful tool to social-ecological planning and design of specific conservation strategies for the sustainable development of the area.

Keywords: social-ecological systems; ecosystem services valuation; land use changes; protected area management; spatial coincidence of biophysical-economic assessments

1. Introduction

Land use changes are the main direct drivers affecting the landscape, causing serious environmental and socioeconomic problems [1,2]. It is considered that they will have the largest global impact on biodiversity by the year 2100 [3], contributing to ecosystems degradation, undermining the ecosystem function and resilience, and thus, threatening the ability of ecosystems to continuously supply services [4]. Land use changes, if anything, are even more worrying in social-ecological systems or cultural landscapes, closely linked to and affected by the socioeconomic structure and cultural aspects.

Social-ecological systems (SES), as cultural landscapes, are the result of the interrelationship between ecosystems and human activities shaped over time [5], conditioned by management policies to take advantage of their spatial heterogeneity and biodiversity [6,7]. Social and ecological

systems coevolve such that ecological systems reflect the characteristics of social systems like values, knowledge, social organization, and technologies, whereas social systems reflect the characteristics of the environment such as species, variation, or resilience [8]. The result is that there are no social systems without nature and few ecosystems without people [9]. Therefore, the adaptation of conservation policies and the application of new approaches to these complex systems are essential [10]. The conservation of cultural landscapes is very relevant within the framework of the European Landscape Convention and its implementation, which has allowed recognizing important conceptual and spatial links between European cultural landscape and the Protected Area network, favoring the resilience of social-ecological systems in the rural areas of Europe [11,12]. However, the method of exploiting ecosystems and the conflict of interest between stakeholders has led to a “conservation against development model” [13–15], affecting both the functionality of natural ecosystems and the sustainability of the coupled human and natural systems [16]. The concept of development has been usually associated with the maximization of the supply of those ecosystem services (ES) with the highest direct market values, basically provisioning services, which likely produce spatial and composition homogenization of the landscape and lead indirectly to the decline of the delivery of other regulating or cultural services [17,18]. Specifically, these last services are not captured by the market and/or are not adequately quantified in comparable terms with economic services and products [19,20], implying their over-exploitation or damage at low or no cost. Hence, given the lack of consideration of the contributions that all types of ES deliver for human well-being, it seems necessary not only to quantify or map the supply of the services provided by the ecosystem, but to combine the biophysical assessment of ES with economic and socio-cultural valuation, so that they can be considered in decision making regarding their conservation [21].

Certainly, the ES framework has made visible the necessity for considering not only ecosystems (supply-side), but also social systems, resulting from the needs and interests of human societies (demand-side). Biophysical assessment is related to the supply-side of ES and is assessed through biophysical indicators. Economic valuation involves the actual allocation of human resources to obtain or enjoy ES, and socio-cultural valuation refers to the relevance given to the ES by stakeholders in a specific area for their own well-being [22]. All of them provide useful information to decision makers about the biophysical components and properties, preferences, demands, or interests of society.

Sitas et al. [23] highlighted the importance of recognizing human demands in decision-making for sustainable development, while Magliocca et al. [24] and Wang et al. [25] suggested that policy-makers must take into account the local situation and changes of land use for the sustainability of the landscape. Actually, the future capability of ecosystems to provide services is determined by socioeconomic and cultural changes, land use, and biodiversity, among others [26].

In environmental policy and decision-making circles, in particular, it has become essential to think about the human economic benefits derived from well-functioning ecosystems. Since the publication of Costanza et al. [19] and *The Economics of Ecosystems and Biodiversity* [27], the study of the economic valuation of ES has increased steadily [28,29] and countless studies have assessed the monetary value of the ES provided, especially by protected areas [14,30–33]. Its importance lies in that it can help to reflect the externalities (positive or negative) of environmental goods at the market price, and therefore, serves as a tool to quantify the trade-offs among different management options [34]. Likewise, it allows the development of specific policies, land use plans, ES payments, and common asset trusts [35]. Thus, the integration of economic and biophysical assessment of ES can inform decision-makers about conservation–conversion trade-offs of landscapes [36,37] by considering environmental and socioeconomic factors. In addition, it may be a suitable tool to mirror their importance and based on this valuation, to make recommendations to develop sustainable policies focused on the SES.

In this sense, with the recognition of the existence of plural values of ES [38,39], the integration of ES values (biophysical, monetary, and socio-cultural) into policy and decision making practices has been a very common matter of debate [40,41]. Recently, plenty of studies have highlighted the importance

of the integrated valuation of ES for landscape planning [42–45]; however, operationalization of biophysical-economic assessment to territorial planning is still not frequent.

In this context, this paper aims to contribute to the existing research by proposing a methodological approach to integrate biophysical and economic assessments of ES in order to provide useful information for the formulation of integrated territorial policies addressed towards sustainable development. The study has been developed in the Urdaibai Biosphere Reserve (UBR), in the Basque Country (Spain), considering that the main objective of biosphere reserves is to reconcile nature conservation with the sustainable use of natural resources, increasing human well-being [46]. In addition, the zoning process, in core, buffer and transition areas, must ensure that biosphere reserves effectively combine conservation and sustainable use of resources. For that reason, they are interesting places to understand the interactions between humans and nature. Our hypothesis is that changes in land use/land cover (LULC) that occurred in the UBR over the last four decades could have had negative consequences on the monetary value of the supplied ES and that their biophysical and monetary values might not coincide in each of the spatial sectors designed in the reserve zoning process. To verify this, we analyzed: (i) The dynamics of LULC and its possible effect on the monetary value of ES and (ii) the spatial coincidence of the biophysical and economic assessments in the delimited zoning units.

2. Study Area

The SES of the Urdaibai estuary, located in Northern Spain (Biscay, the Basque Country) (Figure 1), with an area of c. 22,000 ha, was designated in 1984 by UNESCO as a biosphere reserve. The zoning scheme of biosphere reserves include a core zone of strictly protected ecosystems, a buffer zone where human activity is limited, and a transition area that extends to the outside area where greater activity is allowed considering the interests of different stakeholders such as local communities, management agencies, or scientists. The UBR is a typical agrarian mosaic landscape that is distinguished by smallholdings and multiple land uses, which over time have maintained a viable and ecologically sustainable production system conserving a high level of landscape diversity and complexity [47]. It is characterized by its high biophysical and cultural value in which nature and society have co-evolved throughout the years conforming a resilient SES [48]. The designation of the biosphere reserve was based on the coastal ecosystems, marshlands and green oak forests (the core area of the reserve) [49]. However, with the beginning of the industrialization, native forests were almost eliminated because of timber and charcoal demand, and many farms were abandoned. To counteract, fast-turnover plantations were expanded thanks to policies that encouraged subsidized planting of *Pinus radiata* and *Eucalyptus* sp. monocultures (30- and 10-year rotation, respectively) [50]. Consequently, the social-ecological resilience of the study area has been threatened [48] and the landscape diversity and complexity have been reduced with serious ecological consequences [47,48], altering its capacity to supply ES [51].

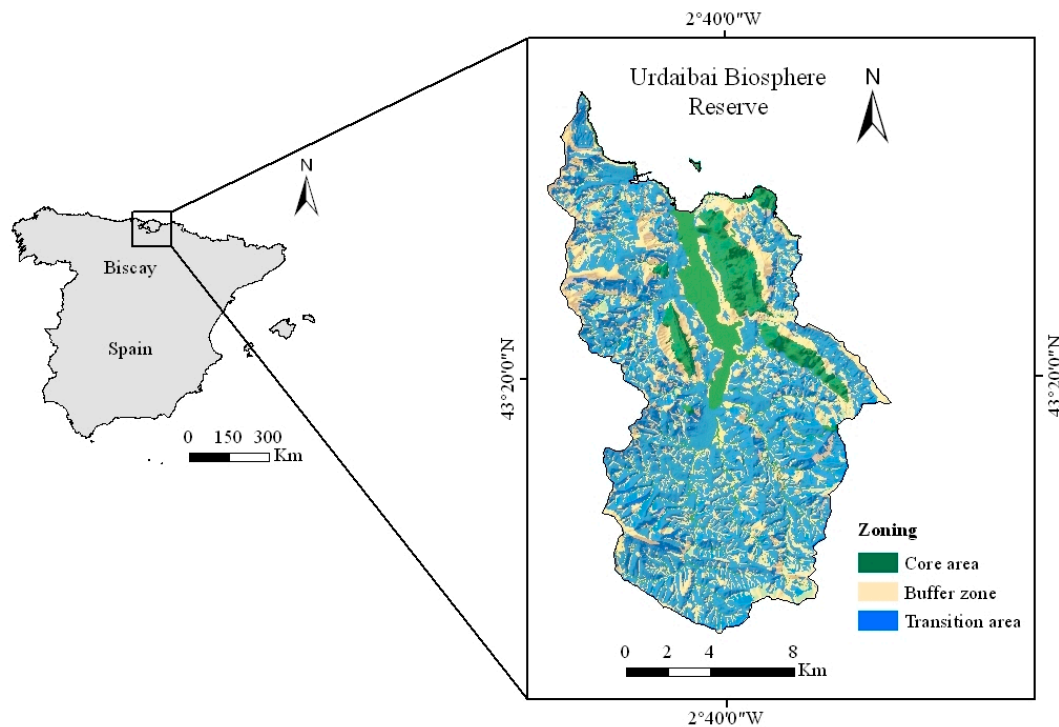


Figure 1. Location of Urdaibai Biosphere Reserve (UBR) in Biscay (Spain). To achieve an integrated management the UBR is divided into three interrelated zones: core area, buffer zone and transition area.

3. Methods

3.1. Selection, Classification and Dynamic of Land Use/Land Covers

We selected and classified land uses into eight common major LULC categories by using high-resolution aerial photos (paper and digital format) of the landscape from 1965 and 1983, as well as available cartography for 2009. These time interval capture different relevant years related to LULC changes tendency: 1965 was the year that changes in economic activity from agriculture and livestock farming to forestry began, 1983 was the year that preceded the designation of the area as a biosphere reserve, and 2009 corresponds to the year of the last update of the habitat EUNIS (European Nature Information System) map in The Basque Country [52], linked to the European Union Habitats Directive [53].

For each year, we generated a grid system of overlapping polygons of approximately 400 m² in size, which was the smallest area we were able to identify in the case of aerial photos from 1965, and assigned them a dominant LULC category in accordance of the majority LULC. The LULC classification included those habitats we considered the most representative, previously used in other studies [19,35,48], and which therefore, best fit for subsequent valuation. This classification was divided in: Croplands, natural and semi-natural grasslands, coastal system, native forest, scrubland, pine and eucalyptus plantations, clear cuts, and urban areas.

3.2. Selection and Valuation of Ecosystem Services

We selected a total of nine ES due to their relevance for conservation planning and socioeconomic development of the area, including three provisioning (food production, timber, and fresh water), four regulating (carbon storage, water regulation and purification, pollination and biological control, and habitat for species) and two cultural services (tourism and recreation and aesthetic enjoyment) (see Supplementary Materials Table S1).

We estimated the biophysical and monetary value of these ES for the year 2009 and integrated both assessments to analyze coincidences across zones in the biosphere reserve. The reason of choosing

2009 year was because indicators used for the biophysical assessment were based on LULC categories, and accurate areas for 1965 and 1983 were not available. In doing so, we used the software ArcGIS 10.4 [54] and established the LULC categories by reclassifying the habitat EUNIS map. Appendix A Table A1 shows the LULC categories identified in the study area.

The use of LULC as suitable proxies is a useful and frequent approach when ES cannot be quantified because of the lack of empirical information about them and how they are changing over time [55–57]. This approach is appropriate in areas, as the one studied here, where primary data and environmental information are absent or lacking [58,59].

3.2.1. Estimation of the Biophysical Value of Ecosystem Services

Due to the difficulty in measuring ES, a key phase in the valuation procedure is the selection of suitable indicators [60]. We used biophysical proxies to assess the capacity of the different LULC categories identified to provide ES (see Supplementary Materials Table S2).

To estimate food production, we added the relative values of the supply of food from agriculture, livestock and fishing, which were in turn calculated on the basis of the average yield of the main crops and livestock, and weight of fish, respectively [61,62]. Fresh water was determined through the SIMPA (Simulation Precipitation-Contribution) model [63] and by using a runoff water proxy, as follows:

$$FW = P - EV \quad (1)$$

where FW is the annual water flow (mm year^{-1}), P is the annual rainfall (mm year^{-1}), and EV is the annual evapotranspiration (mm year^{-1}). Annual evapotranspiration was calculated by using potential and real evapotranspiration maps (1980/81–2005/06 periods) [64]. However, as the potential evapotranspiration does not take into consideration the scarcity of water in soil or vegetation impact, and in order to calculate a more realistic value for the evapotranspiration, we applied a correction factor for the different vegetation types from the InVEST—Integrated Valuation of Ecosystem Services and Tradeoffs [65] and rescaled the values to the real evapotranspiration of the Oka river basin.

Carbon storage supply was assessed following the same procedure of Onaindia et al. [51], on the basis of the C stored in living biomass and soil. C stored in soil was estimated through the “Inventory of organic C stored in the first 30 cm of the soil” of the Basque Country [66], whereas C stored in living mature trees was calculated as follows [67]:

$$CB = V \text{ BEF} (1 + R) D \text{ CF} \quad (2)$$

where CB is the carbon stocks in living biomass (t C ha^{-1}); V is the merchantable volume data for each tree species ($\text{m}^3 \text{ ha}^{-1}$), according to the Forest Inventory of the Basque Country 2011 [68]; BEF is the biomass expansion factor for the conversion of merchantable volume to aboveground tree biomass [69]; R is the root-to-shoot ratio to include belowground tree biomass; D is the basic wood density (t d.m.m^{-3}), for the Northern Iberian Peninsula forests [70,71], and CF is the carbon fraction of dry matter, t C.

We estimated the supply of water regulation and purification by using the water retention index developed by Maes [72] through ESTIMAP (Ecosystem Services Mapping) model. To quantify pollination and biological control, we estimated a pollination index of the likely abundance of pollinator species in each ecosystem by using the InVEST software [65]. To estimate habitat for species, we also followed the procedure of Onaindia et al. [51] and added native plant richness, habitat quality (successional level), and legal protection. Finally, we used recreation and landscape aesthetic indices estimated by Peña et al. [73] to quantify the supply of tourism and recreation and aesthetic enjoyment, respectively.

Every ES was mapped and, using zonal statistics tool in ArcGIS 10.4 [54], mean and standard deviation of each ES provided by different LULC category in the three zones of the UBR were quantified. The spatial measurement units were ha.

3.2.2. Estimation of the Monetary Value of Ecosystem Services

We estimated the monetary value of ES (ESV) for each LULC category identified in the UBR as follows [74]:

$$ESV = \sum_{k=1}^n A_k VC_k \quad (3)$$

where ESV refers to the total value of the ecosystem service function; A_k is the area (ha) of each LULC category, k , and VC_k is the monetary value coefficient (Int.\$ ha⁻¹yr⁻¹) for each LULC category, k .

To the best of our knowledge, there are no previous studies that economically value ES for LULCs in the UBR or other region next to it. Thus, we based the economic valuation method on the benefit transfer approach. This methodology involves estimating the monetary value of ES (ESV) for one context by utilizing results from existing studies [75,76]. Although the limits of the benefit transfer method have been greatly debated in environmental economics literature [76–78], this approach is gaining popularity to inform about nonmarket ES values in a manner relevant to the timeframe and budget within which decisions often have to be made [76]. Actually, it is often considered a valuable tool to provide useful information while reducing costs, constituting a simple tool that avoids the complexity of other methods and allows gaining in understanding, comparability, fast and easy calculation and feasibility when the time and budget are limited [79,80]. Yet, its practicality may be highly reduced because of the lower level of validity and reliability of transferred value estimates [81].

Aware of the importance of sites similarities when transferring monetary values, if possible, literature from close areas with similar characteristics was used. The VC of fresh water provision, water regulation and purification, and habitat for species (except for coastal systems) were estimated according to the data available from the project Valuation of Spanish Natural Assets [82]. In this project, VC of fresh water and water regulation and purification was estimated in accordance with the SIMPA model and water consumption and avoided costs of inland water treatment, respectively; while Nature 2000 management costs for conservation were used for habitat for species.

ESV carbon storage was estimated from the product of the carbon tons stored in living biomass and soil in each type of LULC for its monetary value marketed in the European Energy Exchange (EEX). The EEX is the largest regulatory framework for emissions trading in the world and gives an average settlement price for 2013–2017 period of 5.75€ t⁻¹CO₂ or 21€ t⁻¹C [83]. These carbon market prices are usually applied for accounting purposes [84]. However, it needs to be considered that estimates are strongly affected by the changes in carbon market prices. Besides, if social cost of carbon, understood as the long-term damage done by 1 t CO₂ emission or the damages avoided for a CO₂ reduction, is being considered, we acknowledge that our value differs substantially. The social cost of carbon has an average value of 36€ t⁻¹CO₂ [85] or 27€ t⁻¹CO₂ [86], according to a meta-analysis of 232 and 578 estimates and a 1% and 3% pure rate of time preference, respectively. Nevertheless, worldwide carbon prices range from less than 1€ t⁻¹CO₂ to 120€ t⁻¹CO₂ and about three quarters of the covered emissions are priced at less than 9€ t⁻¹CO₂ [87]. In addition, the willingness to pay for maintaining climate and air regulation in the UBR corresponds to less than 1€ t⁻¹CO₂ [88], which, together with the entirely commercial target of tree plantations in the UBR, leads us to think that the value obtained from the EEX is reassuringly reasonable.

Similarly, estimates for timber provided by native forest and pine and eucalyptus plantations were determined on the basis of the prices perceived by farmers (2007–2011 years) and authorized wood-cuts. Then, we used the averaged values developed by Costanza et al. [35], which derived from the values used in de Groot et al. [4] and the Ecosystem Service Valuation Database [89], containing more than 1350 data-points from over 300 case studies with different valuation methods. These studies are the most complete and reliable for a first approximation of economic quantification and have been widely applied in estimating ecosystem services values all over the world [74,90,91]. We were careful to choose monetary values calculated for European countries, similar to our area in terms of their ecological, socio-cultural and economic characteristics. Still, some of them were not available and

we had to use values for other continents. Besides, there is a lack of data for some ES and LULC, so other sources, as similar as possible to our study area, were needed to complete the values. Lastly, in spite of classifying urban areas and although the provision of ES by urban green infrastructures is well-known [92–94], we excluded them from the valuation process for two main reasons: First, the territorial scope of the Plan for Use and Management, basic instrument to manage the area of the biosphere reserve, is only focused on unbuilt lands [49]; second, our spatial analysis was not precise enough to distinguish green spaces in urban areas. Likewise, clear cuts neither were valued assuming that their value is likely negligible because this type of management involves the temporal destruction of the ecosystem.

All of the obtained values were adjusted to 2009 Euros per hectare per year so as to make possible the comparison of the values, which were assessed in different units. Following the procedure applied in de Groot et al. [4], if necessary, we converted the estimates into Euros by using official exchange rates and then, adjusted to 2009 values using the GDP deflators [95]. Appendix A Table A1 shows the most representative biomes used as proxies for the LULC categories. Supplementary material Table S3 displays the details of all data used for the benefit transfer and the VC of ES.

Yet, no coefficient of sensitivity (CS) was applied to analyze the sensitivity and robustness of VC. In fact, despite the CS of Mansfield [96] is widely used in many ES studies [76,97–100]), there is large uncertainty and lack of validity regarding this approach. According to this analysis, if $CS < 1$, then the estimated ecosystem value is inelastic in relation to VC and the results of the ESV calculations are reliable even if the VC has relatively low accuracy, and vice versa. Nonetheless, the way they are calculated, CS values will always be in the range between 0 and 1, and besides, they will be always positive, which is not realistic in a real market either [101].

Lastly, we assessed the impact of LULC changes in the ESV by calculating the differences among monetary values in the time series considered from 1965 to 1983 to 2009, according to the areas of the dominant LULC categories, estimated through high-resolution aerial photos of the landscape. Following the same procedure as Zhao et al. [90], Li [102] or Song and Deng [103], we based on the 2009 value of the Euro and did not adjust monetary values to the corresponding dates to simplify comparison and avoid problems with inflation and deflation.

3.3. Integrating Biophysical and Economic Assessments of Ecosystem Services. Spatial Adjustment

We evaluated the degree to which economic valuation of ES coincides with the biophysical assessment of ES, or vice versa, by considering the zoning schemes of biosphere reserves, i.e., core area, buffer zone, and transition area. In doing so, we analyzed the spatial coincidence between both assessments by simply juxtaposing the relative levels of both assessment (i.e., low, medium, high) for each LULC category and zone, as shown in Figure 2. Specifically, we based on two data sets: (1) A biophysical vector, whose elements (a_{mx1}) are the normalized biophysical values of each zone of the UBR. Since ES biophysical values have different dimensions, to allow their comparison, each ES was scaled individually from 0 to 1 for each LULC category and zone, according to their minimum and maximum values (5); and (2) a monetary vector, whose elements (b_{mx1}) are the monetary values of each zone of the UBR.

$$x_{\text{normalized}} = \frac{x - \min(x)}{\max(x) - \min(x)} \quad (4)$$

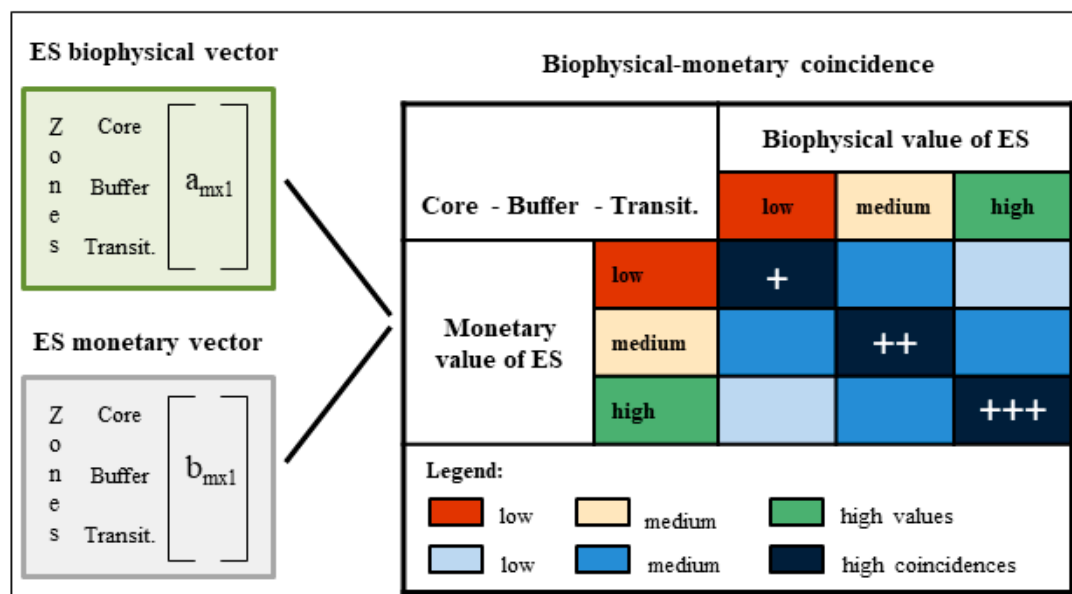


Figure 2. General overview of the steps followed for the estimation of the biophysical-monetary coincidence, where a_{mx1} and b_{mx1} are the normalized biophysical and monetary values of the biophysical and monetary vectors, respectively. The table of biophysical-monetary interactions shows the three categories of ES biophysical and monetary values in each zone, classified in three levels (low, medium and high values) and every possible spatial coincidence between them (low, medium and high coincidences). In the figure, the darker the blue color the higher the coincidence values. High coincidences were also subdivided, considering low (+), medium (++) and high (+++) degrees of adjustment Transit. = transition area. Note: For a similar cross-tabulation, see Schmitz et al. [104], Burkhard et al. [105], or Albert et al. [106].

4. Results

Biophysical and monetary values of ES, as well as changes in the monetary value of ES throughout the years, are presented below, to subsequently evaluate the spatial coincidence between biophysical and economic assessments across the core, buffer, and transition areas of the UBR.

4.1. Biophysical Value of ES of the Zoning Units

Table 1 shows the normalized biophysical values obtained from the mapping procedure for each ES and LULC category and zone, and their added values by category of services. If we consider the whole of the biosphere reserve, native forest was the LULC with the highest contribution to ES per hectare, followed by scrublands and croplands, while a coastal system apparently contributed the least to ES (Table 1). By category of services, croplands were the main providers of provisioning services, whereas native forest and coastal system contributed the most to regulating services and cultural services, respectively (Table 1). As for the lowest biophysical values, coastal system had the lowest ones in both provisioning services and regulating services, and pine and eucalyptus plantations were the ones with the lowest ones in cultural services (Table 1). However, when zoning was only considered, we found that the core area and buffer zone presented the highest values, while the transition area seemed to have the lowest values. Still, differences among zones were not especially relevant, and overall, it could be said that ES are well-distributed across every zone, except for the supply of cultural services that was higher in the core area (Table 1).

Table 1. Normalized biophysical values obtained from the mapping procedure for each ES and LULC category and zone, and their added values by category of services. Measurement units are indicated in brackets.

	Core Area						Buffer Zone						Transition Area						UBR					
	CR	GR	CO	NAT	SCR	PI	CR	GR	CO	NAT	SCR	PI	CR	GR	CO	NAT	SCR	PI	CR	GR	CO	NAT	SCR	PI
Provisioning services (total)	1.93	0.93	0.90	1.24	0.91	1.86	1.63	0.92	1.01	1.34	0.96	1.76	1.61	0.94	0.01	1.87	0.96	1.75	1.56	0.73	0.39	0.58	1	1.12
Food production (t/ha)	1	0.01	0.01	0	0	0	0.75	0.01	0.01	0	0	0	0.69	0.01	0.01	0	0	0	1	0.01	0.01	0	0	0
Timber (m ³ /ha)	0	0	0	0.39	0	0.98	0	0	0	0.49	0	0.89	0	0	0	1	0	0.88	0	0	0	0.58	0	1
Fresh water (mm/ha)	0.93	0.93	0.89	0.86	0.91	0.89	0.88	0.92	1	0.86	0.96	0.87	0.92	0.93	0	0.87	0.96	0.87	0.56	0.72	0.38	0	1	0.12
Regulating services (total)	1.20	1.92	0.98	3.39	2.60	2.49	1.94	2.10	0.93	3.17	2.53	2.34	1.66	2.10	0.57	2.95	2.56	2.37	1.46	1.94	0.55	3.54	2.62	2.66
C storage (t C/ha)	0.31	0.26	0	0.78	0.27	0.71	0.47	0.28	0.19	0.80	0.31	0.73	0.42	0.26	0.06	1	0.31	0.72	0.31	0.05	0	1	0.13	0.83
Water regulation and purification (-)	0.58	0.54	0.42	0.63	1	0.87	0.78	0.67	0.09	0.65	0.80	0.77	0.66	0.73	0	0.77	0.82	0.91	0.57	0.66	0	0.54	0.90	1
Pollination and biological control (-)	0.30	0.82	0	1	0.77	0.61	0.54	0.83	0.12	0.71	0.79	0.55	0.47	0.80	0.06	0.63	0.78	0.45	0.57	0.97	0	1	0.95	0.59
Habitat for species (-)	0	0.32	0.56	0.98	0.57	0.30	0.15	0.32	0.53	1	0.64	0.29	0.11	0.31	0.45	0.54	0.64	0.29	0	0.25	0.55	1	0.64	0.23
Cultural services (total)	1.06	1.25	1.71	1.28	1.59	0.53	0.62	0.93	1.65	1.19	1.10	0.33	0.41	0.64	1.59	0.17	0.87	0	0.43	0.78	2	1.26	1.21	0
Tourism and recreation (-)	0.49	0.67	0.71	0.65	0.85	0.25	0.15	0.39	0.77	0.55	0.51	0.13	0.07	0.30	1	0.14	0.45	0	0.09	0.43	1	0.74	0.70	0
Aesthetic enjoyment (-)	0.57	0.58	1	0.62	0.75	0.28	0.47	0.54	0.88	0.64	0.60	0.20	0.34	0.35	0.59	0.03	0.42	0	0.33	0.35	1	0.52	0.51	0
Total	4.19	4.11	3.59	5.91	5.11	4.89	4.20	3.95	3.59	5.70	4.60	4.42	3.68	3.68	2.17	4.98	4.39	4.12	3.45	3.45	2.94	5.38	4.82	3.78
	27.79						26.46						23.02											

Note: Since ES units are different, and consequently it was not possible their comparison, each ES was scaled individually for each LULC category and zone, according to their minimum and maximum and compared as relative values. CR = Cropland, GR = Natural and semi-natural grassland, CO = Coastal system, NAT = Native forest, SC = Scrubland, PI = Pine and eucalyptus plantations.

4.2. Monetary Value of ES

4.2.1. Monetary Value of ES of the Zoning Units

Table 2 shows the monetary values obtained for each ES and LULC category and zone according to their relative area, and their added values by category of services. Overall, the transition area presented the highest monetary values per hectare (>9000 €/ha/yr), followed by the core area (almost 7000 €/ha/yr) and the buffer zone (>5600 €/ha/yr) (Table 2). The highest monetary values for provisioning and regulating services were found in the transition area, croplands (1362.2 €/ha/yr), and pine and eucalyptus plantations (2241.4 €/ha/yr), respectively, whereas coastal system and native forest of the core area (547.8 and 317.8 €/ha/yr, respectively) had the highest values for cultural services (Table 2). Zone by zone, coastal systems were the most valued LULC in the core area (3721.8 €/ha/yr), followed by croplands in the transition area (3106.4 €/ha/yr) and native forest in the buffer zone (2307.5 €/ha/yr) (Table 2).

4.2.2. Changes in the Monetary Value of ES

Table 3 shows the total monetary value of ecosystem services estimated for each LULC category and category of ES. As suggested by the results obtained, the LULC changes that occurred in the study period (Appendix A Table A2) seem not to impact in any way the ESV, being the global monetary value almost the same as in the past (Table 3). In fact, while the ESV of regulating services seemed to increase 58×10^5 €, provisioning services and cultural services reduced their ESV almost 56×10^5 and 1×10^5 €, respectively. This change in ESV was mainly due to the abandonment of multiple agricultural activities and its transformation to predominance of forestry activity (pine and eucalyptus plantations). The decrease in the agricultural area (-67%) led to a reduction of the food production service, concretely, the most valued service in monetary terms, so the impact on the global ESV was quite noteworthy, even if regulating services ESV presumably increased due to the high value of carbon storage. Likewise, although from 1965 to 1983, the area of the natural and semi-natural grasslands and the native forest decreased by 35.8% and 14.4%, respectively, from 1983 to 2009, with the expansion of clear cuts, natural and semi-natural grasslands were expanded by 88.6%, and native forest increased by 15% its area, resulting in an increase of 34.7×10^5 and 22.3×10^5 €, respectively (Table 3). The coastal system has remained practically unchanged in the area throughout the study period, but scrubland has been drastically reduced (-73.3%), representing losses of 25.7×10^5 €. Finally, in spite of not being valued economically, it is important to point out that urban areas demonstrated the largest increase growing more than 200%, especially during the last period between 1983 and 2009, though their coverage represents only 2.2% of the total area (Appendix A Table A2).

Table 2. Monetary values obtained for each ES and LULC category and zone according to their relative area, and their added values by category of services (ESV in 2009 €/ha/yr).

	Core Area						Buffer Zone						Transition Area					
	CR	GR	CO	NAT	SCR	PI	CR	GR	CO	NAT	SCR	PI	CR	GR	CO	NAT	SCR	PI
Provisioning services (total)	204.3	59.0	1224.4	122	15.7	18.1	296.4	201.7	7.7	121.1	105.5	114.8	1362.2	791.3	5.3	37.4	74.7	243.6
Food production (t/ha)	189.1	49.6	1224.4	0	0	0	274.3	169.6	7.7	0	0	0	1260.5	665.4	5.3	0	0	0
Timber (m ³ /ha)	0	0	0	19	0	9.5	0.0	0.0	0	18.9	0	60	0.0	0.0	0	5.8	0	127.4
Fresh water (mm/ha)	15.2	9.4	0	103	15.7	8.6	22.1	32.1	0	102.2	105.5	54.7	101.6	125.9	0	31.5	74.7	116.2
Regulating services (total)	224.3	76.3	1949.7	1884.4	123.2	166.7	325.4	260.9	12.3	1870.8	828.4	1056	1495.6	1023.9	8.4	577.4	586.9	2241.4
C storage (t C/ha)	217.9	71.9	1128	1667	118.9	163.1	316.0	245.9	7.1	1654.9	799.3	1032.8	1452.4	965.0	4.9	510.7	566.3	2192.1
Water regulation and purification (-)	0.7	0.6	103.7	3.4	0.7	0.4	1.0	1.9	0.7	3.4	4.4	2.6	4.8	7.5	0.4	1	3.2	5.6
Pollination and biological control (-)	4.5	2.7	527.1	151.4	1.8	2	6.5	9.4	3.3	150.3	12.4	12.4	29.8	36.8	2.3	46.4	8.8	26.4
Habitat for species (-)	1.3	1.1	190.9	62.7	1.8	1.3	1.9	3.7	1.2	62.2	12.3	8.1	8.6	14.5	0.8	19.2	8.7	17.3
Cultural services (total)	37.3	8.2	547.8	317.8	0.4	12.9	54.1	28.2	3.5	315.5	2.8	81.7	248.7	110.5	2.4	97.4	2	173.5
Tourism and recreation (-)	6.7	1.3	516.2	317.5	0.4	12.9	9.7	4.4	3.3	315.2	2.8	81.7	44.5	17.3	2.2	97.3	2	173.5
Aesthetic enjoyment (-)	30.6	6.9	31.6	0.3	0	0	44.4	23.8	0.2	0.3	0	0	204.2	93.2	0.1	0.1	0	0
Total	466.0	143.5	3721.8	2324.3	139.4	197.7	675.9	490.8	23.5	2307.5	936.7	1252.5	3106.4	1925.7	16	712.1	663.7	2658.5
	6992.7						5686.9						9082.5					

Note: See that if monetary values for each ES and LULC are added, the obtained value would be exactly the same as in Table S2, i.e., VC. CR = Cropland, GR = Natural and semi-natural grassland, CO = Coastal system, NAT = Native forest, SC = Scrubland, PI = Pine and eucalyptus plantations.

Table 3. Total monetary value of ecosystem services (ESV in 2009 € × 10⁵/yr) and percentage (in brackets) estimated for each LULC category and divided by their three main categories according to CICES [107].

LULC	1965				1983				2009				Changes		
	Prov.	Reg.	Cult.	Total	Prov.	Reg.	Cult.	Total	Prov.	Reg.	Cult.	Total	1965–1983	1983–2009	1965–2009
CR	111.9	122.9	20.4	255.2 (30.1%)	83.1	91.3	15.2	189.6 (21.6%)	36.9	40.5	6.7	84.1 (9.9%)	−65.6	−105.5	−171.1
GR	25	32.4	3.5	60.9 (7.2%)	16.1	20.8	2.2	39.1 (4.5%)	30.3	39.2	4.2	73.8 (8.7%)	−21.8	34.7	12.9
CO	11.2	17.8	5	34 (4%)	11.2	17.8	5	34 (3.9%)	11	17.4	4.9	33.3 (3.9%)	0	−0.7	−0.7
NAT	9.1	140.7	23.7	173.5 (20.4%)	7.8	120.4	20.3	148.4 (16.9%)	9	138.4	23.3	170.7 (20.1%)	−25	22.3	−2.8
SC	3.9	31	0.1	35 (4.1%)	1.4	10.7	0	12.1 (1.4%)	1.1	8.3	0	9.4 (1.1%)	−23	−2.7	−25.7
PI	26.6	244.8	18.9	290.3 (34.2%)	41.7	383.7	29.7	455.1 (51.8%)	43.9	403.5	31.2	478.6 (56.3%)	164.8	23.5	188.3
Total	187.8	589.4	71.7	848.9	161.2	644.6	72.5	878.2	132	647.4	70.5	849.9	29.4	−28.4	1

Prov. = Provisioning services; Reg. = Regulating services; Cult. = Cultural services. CR = Cropland, GR = Natural and semi-natural grassland, CO = Coastal system, NAT = Native forest, SC = Scrubland, PI = Pine and eucalyptus plantations. Note: Differences in the monetary value may be found when comparing to the values obtained for 2009 in Table 2, especially regarding cropland and natural and semi-natural grasslands. These differences are due to bias in LULC occupied areas, estimated through aerial photos and dominant LULC category, as explained above.

4.3. Spatial Zoning Coincidence between Biophysical and Economic Assessments

Figure 3 shows the degree of adjustment between the biophysical and economic assessments of ES into the core, buffer and transition areas of the UBR. There were not found high coincidences between the two assessments in none of the delimited zones. The coincidence across buffer and transition areas, with high biophysical-low monetary values and low biophysical-high monetary values, respectively, was low, being the core area and the only one presenting medium coincidences.

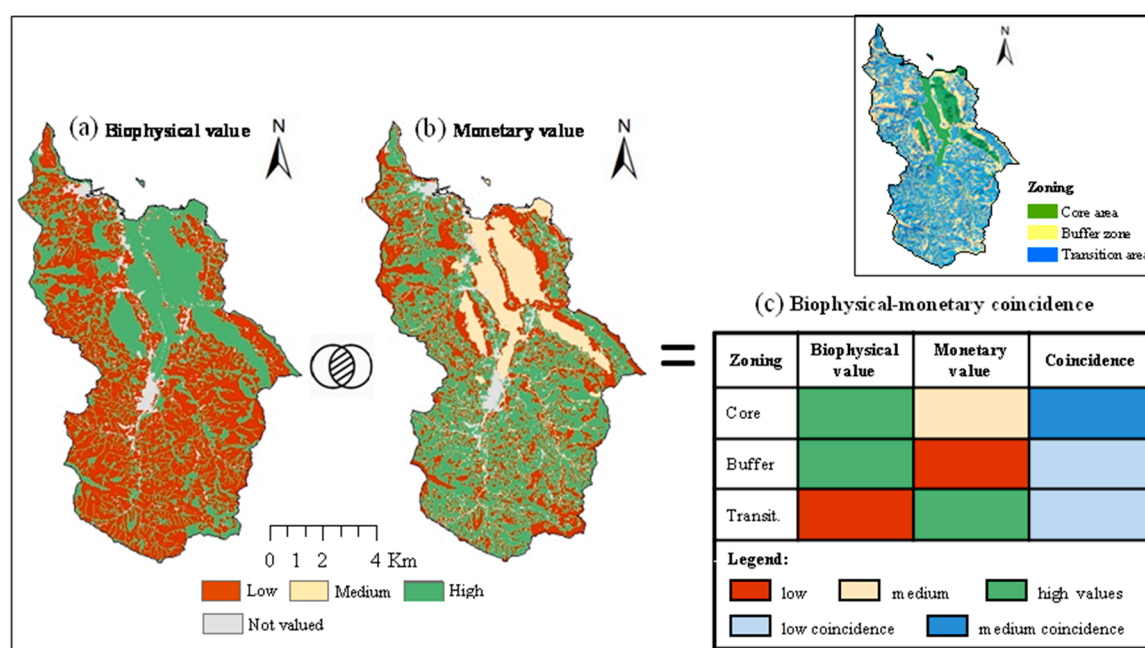


Figure 3. Mapping of both biophysical values of ES (a) and monetary values of ES (b) for each zone. Values were ranked on low-medium-high equal scale from green to yellow to brown. (c) Biophysical-monetary coincidence across the core, buffer and transition areas of the UBR. Urban areas were not evaluated (light grey zones).

5. Discussion

Several studies state that unsustainable practices like intensive farming and fishing, urban development and high-intensity logging as referred to here, not only have a negative ecological impact, but also as regards human preferences, and ultimately, economy [20,91,103,108]. However, the LULC changes occurred in the UBR during the last 44-year period, characterized principally by the increase of unsustainable logging practices, have not revealed any decrease in the ESV, but have maintained constant in time and even increased slightly.

The biggest decrease observed was mainly due to the abandonment of croplands from 1983 to 2009, which was one of the most economically valued LULC. Actually, although a Strategy of Sustainable Development was developed in 2009–2012 in the Reserve to enhance the sustainable ecological agriculture [109], these objectives have not been achieved yet. In contrast, the area of coastal system and native forest has remained fairly constant, so does the ESV. Specifically, as for the native forests, there was a decrease in the first period due to the replacement by pines and eucalyptus plantations, but a significant increase (14.1%) in the second period after the biosphere reserve designation. Similarly, other valuable LULC, such as natural grasslands, have had a similar dynamic. All this could be understood to be a success due to the management rules established by the Reserve Government for the conservation of biodiversity and natural resources [49,110]. Among others, a Territorial Action Plan has been developed for the suitable management of the Cantabrian evergreen-oak forest and their protection zones, LIFE projects (financial instruments supported by the European Union) have been carried out for the regeneration of coastal sands and restoration of

marshlands, and also broad-leaf species have been planted in public lands as well as private lands through cession agreements. Nevertheless, the Governance Plan for Use and Management approved in 1993 and that should have been revised more than 15 years ago, is still a matter of discussion among different social groups. Some reasons to reject the new reviewed version of the Plan [111] included both the criticism by ecologist groups arguing that the protection of the biosphere reserve would be reduced, and the disagreement on the part of the Association of Foresters of Biscay who stated that forestry would be highly limited.

The management of the UBR has only been focused on a few marketed services, such as timber. The point is that carbon storage was, after food provision, the next most valued ES in monetary terms, and consequently, the economic impact of LULC changes over time was relatively low due to the great increase of pine and eucalyptus plantations playing a key role in carbon sequestration. Under this argument, forest managers in the region persist in the cultivation of pine and eucalyptus plantations. However, it must be considered that, once cut, the wood is transformed mainly into pellets for bioenergy, and therefore, in short-term carbon storage products.

Similarly, pine and eucalyptus plantations are also one of the highest suppliers of water regulation and purification. Nevertheless, the provision of this regulatory services is negatively affected due to forest management (clear-cuts each 30 years for pines and 10 years for eucalyptus), which leads to numerous externalities or environmental side effects, such as erosion and soil loss [47], the worsening of water quality [112], loss of species diversity, and also, landscape aesthetic aspects [51,113]. These negative environmental externalities are related to economic losses that in this paper have not been considered in the monetary valuation, but should be accounted for in the decision-making processes. In addition, these processes have promoted landscape spatial homogenization decreasing its multifunctionality.

As a consequence, the transition area, with the least restrictions of use and the highest forestry activity of the UBR, showed the lowest biophysical values and the highest monetary values. Thus, the spatial zoning adjustment between biophysical and economic assessments showed that transition and buffer zones, presented the lowest values of coincidence, whereas the core area, aimed at enhancing the conservation of natural resources and the rural cultural landscape, had the highest (medium value) degree of valuation coincidence. This greater coincidence highlights the core zone over the others for its social-ecological importance and conservation interest.

Recommendations for Management

In general, the focus of conservation planning is based on the biological benefits. Most of the conservation schemes incorporate economic costs simplistically, assuming that all areas are equally costly, without considering the spatial variability of costs over landscapes and regions [114]. Nowadays, a key challenge in conservation policies is to somehow integrate biophysical and socioeconomic values of ES [42,115]. Even so, the design of management policies does not always take these interactions into account, which can lead to inadequate conservation decisions. In fact, only an integrated approach can explore the linkages of the functioning of ecosystems and socioeconomic values and human well-being [116]. Thus, to achieve more efficiently conservation planning a cost-effectiveness analysis should be performed. So, beyond some methodological caveats already mentioned, this study represents a methodological approach towards the integrated assessment of ES from the biophysical and economic perspectives. The method followed has allowed us to unravel the links between both dimensions in the UBR zoning scheme, in order to provide a useful tool for the future management of the protected area. Additionally, it could also be valuable in identifying the most valued ES for the local community along biophysical and economic assessments.

Our results highlight a gradient of biophysical value according to the degree of conservation in the UBR zoning scheme. In fact, the highest biophysical values of ES were found in the core and buffer zones. However, conservation criteria changes over time and socio-cultural context, so focusing solely on biophysical values will fail in biodiversity protection. In this regard, contrary to

the biophysical value dimension, the transition area had the highest monetary values of ES. Then, when biophysical-monetary coincidence was analyzed, no high coincidences were found in none of the zones, suggesting that each dimension provides divergent, but complementary information on their value.

In this sense, important mismatches, i.e., differences in quality or quantity between biophysical and monetary value dimensions [22], were found in previous studies in the UBR [88], and other protected areas [42,117]. These differences between biophysical and monetary values are usually related to the relative abundance of ES and its derived economic value [118]. That is, a service can be more abundant, and consequently, its monetary value can be high or a service can be scarce or deteriorated, and precisely because of that, its monetary value can be even higher. Therefore, even if benefit transfer method might give insights for ES economic valuation and conservation decision making, it is fundamental to work locally to be more context-specific. Synthetically, and considering the limitations and controversy of economic valuation of ES by itself [119,120], it is important to highlight that it cannot be a substitute for biophysical assessment and it should only be acceptable when management of ecosystems develops synergies between ecological and social systems, leading to an egalitarian social-ecological transition [121,122]. Besides, it would be essential to include other value dimensions like socio-cultural. After all, people's preferences are also motivated by non-economic reasons like ethic or their relationship with nature [123]. Therefore, if the whole complexity of ES is to be captured, we need to integrate biophysical, monetary, and socio-cultural values.

On the other hand, previous studies in the UBR show that designation and management of the biosphere reserve have been positive for nature conservation and life quality local population [110]. The obtained findings here reaffirm the need for new approaches to balance the conservation goals and social requirements. In this social-ecological context, an adaptive management is promising [124]. The procedure developed could give, relatively easily, guidelines to adjust to the different situations over time and help in the redesign of the biosphere reserve in an adaptive manner. Specifically, the buffer zone and transition area, with the lowest coincidences would need to be more promoted so that a sustainable balance between biophysical and monetary values is to be achieved. For example, if we consider that the transition area, with most of its surface covered by pine and eucalyptus plantations, presented the lowest biophysical values, but the highest monetary values, as was suggested by Rodríguez-Loinaz et al. [113], we propose to replace exotic plantations by native forests in those areas with slopes higher than 30% or with erosion risk. In this way, the potential for increasing carbon sequestration in the long-term (>50 years) would be higher and other ES would also be favored, while avoiding the environmental problems the actual plantations cause [113], and consequently, biophysical and economic assessment could be matched. Similarly, the replacement of forestry land with high agrological capacity by the very highly economically valued (sustainable) agriculture would improve landscape multifunctionality, increase self-provisioning, recover natural ecosystems and maintain biodiversity and a diverse flow of ES [125,126], leading to a more balanced biophysical-economical assessment. Likewise, it is fundamental to stress the monetary value of ES provided by coastal systems, which seem not to be sufficiently recognized.

Yet, only a holistic approach to landscape description can explain landscape complexity and trade-offs and synergies between ES [127]. Thus, conserving multifunctional landscapes that provide multiple ES should be a priority. Multifunctionality involves meeting multiple ecological, economic and socio-cultural services for multiple social actors within a territory. Hence, considering that 92% of the territory is privately owned, the involvement of affected local populations is necessary and their ES preferences should also be taken into account, and if necessary, compensated, so that private and public sectors could agree on a type of win-win solution.

6. Conclusions

The LULC changes occurred during the last 44-year period have not had any substantial impact on the economic valuation of ES, but the expansion of pine and eucalyptus plantations have homogenized the territory with a subsequent loss of multifunctionality. However, results obtained from the biophysical value of ES suggest that the establishment of the biosphere reserve may have contributed successfully to its conservation goal, being the core area and the buffer zone the ones with the highest values. Likewise, the transition area, which was economically the most valued area, fulfills its function of promoting sustainable economic activities that encourage the socioeconomic development of the local population. However, negative environmental externalities related to some intensive forestry practices, especially in pine and eucalyptus plantations, should be considered. The methodological approach developed in this study allows exploring the spatial interactions between biophysical and economic assessment of ES, which may be useful for decision-making and design and zoning of the biosphere reserve. In the UBR, this procedure followed allows us to identify that the zoning design of the reserve was performed based on its biophysical values. In fact, no high coincidences were found for the different zoning units of the reserve, which shows evident divergences between the economic and biophysical dimensions. Even so, the core area was the one that had the highest coincidences (medium values) between both ES assessments, which highlights its importance, not only in biophysical terms, but also economically. This method is especially appropriate for its application in social-ecological systems, such as the UBR, since it allows quantifying the interdependencies between biophysical and socioeconomic dimensions, which together with the socio-cultural one, are completely necessary to guarantee a sustainable management. Therefore, it is a useful tool for spatial planning and design of specific conservation policies for the sustainable development of the area, including the assessment of the effectiveness of protected areas.

Supplementary Materials: The following are available online at <http://www.mdpi.com/2071-1050/11/11/3092/s1>, Table S1: Ecosystem services classification used and its equivalence with other classification systems, Table S2: Indicators used for the biophysical assessment of ecosystem services, Table S3: Monetary value coefficients in 2009 €/ha/yr estimated for each LULC category, valuation method and source of the data.

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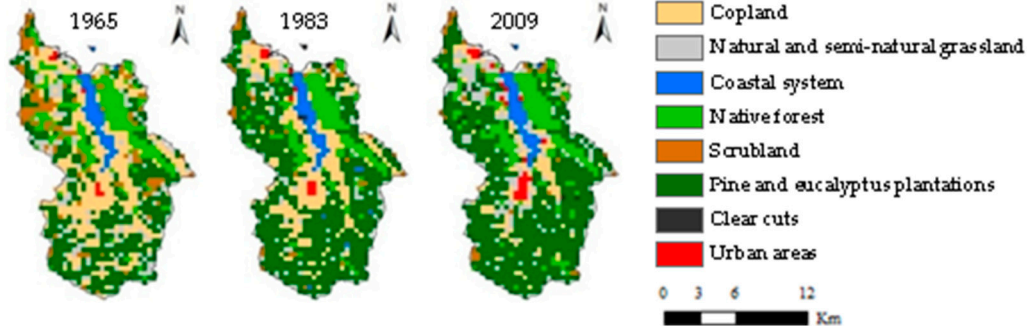
Appendix A

Table A1. LULC categories identified in the study area with their EUNIS habitat type code and the most representative biomes [4] used as proxies. CR = Cropland, GR = Natural and semi-natural grassland, CO = Coastal system, NAT = Native forest, SC = Scrubland, PI = Pine and eucalyptus plantations, CC = Clear cuts, URB = Urban areas.

LULC Category	EUNIS Habitat Type Code	The Most Representative Biome	Description
CR	FB.4, G1.D(X), I1.2	Cropland	Croplands and orchards
GR	E1.26, E2.11, E2.13(Y), E2.21, E3.41, E5.6, H3.2, H5.6	Grasslands	Temperate grasslands, tropical grasslands, boreal grasslands and mountainous grasslands
CO	A2.511, A2.627, A2.636, A2.63C, A2.651, A2.654, A2.658, B1.21, B1.31, B1.32, B1.42, B3.23, B3.31, F4.231, C2.4, C3.21	Coastal systems	Estuaries, sea-grass fields, shallow seas of continental shelves, rocky shores and beaches found in the terrestrial near-shore as well as the intertidal zones
NAT	F9.12(Y), F9.2(X), F9.2(Y), G1.21(Z), G1.62, G1.7B1, G1.86, G1.A1, G1.A1(X), G1.C(X), G1.C(Y), G1.C1, G1.C2, G2.121, G4.(Z), G4.F, G5.61, G5.72, G5.73	Temperate forest	Temperate deciduous forest, temperate broadleaf and mixed forest, temperate coniferous forest, temperate rainforest
SC	E5.31(X), E5.31(Y), F3.11(X), F3.11(Y), F3.15(X), F3.15(Y), F4.23(X), F5.21(Y), F7.44(Y), FA.1, FA.3, I2.3	Woodlands	Shrublands, scrublands, savannas and chaparral interleaved with one another in mosaic landscape patterns
PI	G1.C3, G2.81, G3.F(L), G3.F(M), G3.F(P), G3.F(Q), G3.F(S), G3.F(T), G3.F(U), G3.F(Y), G5.74, G5.81, G5.82	/	Plantations of conifers (basically <i>Pinus radiata</i>) and eucalyptus species
CC	/	/	/
URB	E2.6, I2.1, I2.2, J1, J2, J3.2, J4.1, J4.2, J4.3, J4.5, J4.6, J4.7, J6	/	/

Table A2. LULC changes and their evolution in time. (a) LULC changes in the UBR in 1965, 1983 and 2009 expressed in ha and percentage; (b) Increment of the LULC changes (%) in the three time ranges studied. CR = Cropland, GR = Natural and semi-natural grassland, CO = Coastal system, NAT = Native forest, SC = Scrubland, PI = Pine and eucalyptus plantations, CC = Clear cuts, URB = Urban areas.

(a)								
Years	CR	GR	CO	NAT	SC	PI	CC	URB
1965	6006.9 (27.6%)	2378.5 (10.9%)	902.8 (4.1%)	3246.5 (14.9%)	2013.9 (9.3%)	7066.0 (32.5%)	0 (0%)	156.3 (0.7%)
1983	4461.8 (20.5%)	1527.8 (7.0%)	902.8 (4.1%)	2777.8 (12.8%)	694.4 (3.2%)	11,076.4 (50.9%)	104.2 (0.5%)	225.7 (1.0%)
2009	1979.2 (9.1%)	2881.9 (13.2%)	885.4 (4.1%)	3,194.4 (14.7%)	538.2 (2.5%)	11,649.3 (53.5%)	156.3 (0.7%)	486.1 (2.2%)



(b)								
Years	CR	GR	CO	NAT	SC	PI	CC	URB
1965–1983	−25.7	−35.8	0	−14.4	−65.5	56.8	104.2	44.4
1983–2009	−55.6	88.6	−1.9	15	−22.5	5.2	50	115.4
1965–2009	−67.0	21.2	−1.9	−1.6	−73.3	64.9	156.3	211.0

Note: Pixels in the maps correspond to the predominant LULC category.

References

1. Foley, J.; DeFries, R.; Asner, G.; Barford, C.; Bonan, G.; Carpenter, S.R.; Cha, F.S. Global Consequences of Land Use. *Science* **2005**, *309*, 570–574. [[CrossRef](#)] [[PubMed](#)]
2. Newbold, T.; Hudson, L.N.; Hill, S.L.L.; Contu, S.; Lysenko, I.; Senior, R.A.; Borger, L.; Bennett, D.J.; Choimes, A.; Collen, B.; et al. Global effects of land use on local terrestrial biodiversity. *Nature* **2015**, *520*, 45–50. [[CrossRef](#)] [[PubMed](#)]
3. Sala, O.; Stuart Chapin, F.; Armesto, J.J.; Berlow, E. Global biodiversity scenarios for the year 2100. *Science* **2000**, *287*, 1770–1774. [[CrossRef](#)] [[PubMed](#)]
4. De Groot, R.; Brander, L.; Van Der Ploeg, S.; Costanza, R.; Bernard, F.; Braat, L.; Christie, M.; Crossman, N.; Ghermandi, A.; Hein, L.; et al. Global estimates of the value of ecosystems and their services in monetary units. *Ecosyst. Serv.* **2012**, *1*, 50–61. [[CrossRef](#)]
5. Berkes, F.; Folke, C. *Linking Social and Ecological Systems: Management Practices and Social Mechanisms for Building Resilience*; Cambridge University Press: New York, NY, USA, 1998.
6. De Aranzabal, I.; Schmitz, M.F.; Aguilera, P.; Pineda, F.D. Modelling of landscape changes derived from the dynamics of socio-ecological systems. *Ecol. Indic.* **2008**, *8*, 672–685. [[CrossRef](#)]

7. Rescia, A.J.; Pérez-Corona, M.E.; Arribas-Ureña, P.; Dover, J.W. Cultural landscapes as complex adaptive systems: The cases of northern Spain and northern Argentina. In *Resilience and the Cultural Landscape: Understanding and Managing Change in Human-Shaped Environments*; Plieninger, T., Bieling, C., Eds.; Cambridge University Press: Cambridge, UK, 2012.
8. Norgaard, R.B. *Development Betrayed: The End of Progress and a Coevolutionary Revisioning of the Future*; Routledge: London, UK, 1994.
9. Petrosillo, I.; Aretano, R.; Zurlini, G. Socioecological Systems. In *Reference Module in Earth Systems and Environmental Sciences*; Elsevier: Amsterdam, The Netherlands, 2015.
10. Levin, S.; Xepapadeas, T.; Crépin, A.; Norberg, J.; de Zeeuw, A.; Carl, F.; Terry, H.; Kenneth, A.; Scott, B.; Gretchen, D.; et al. Social-ecological systems as complex adaptive systems: Modeling and policy implications. *Environ. Dev. Econ.* **2013**, *18*, 111–132. [\[CrossRef\]](#)
11. Rescia, A.J.; Schmitz, M.F.; Pineda, F.D. Ecological considerations for planning and management of cultural fragmented landscapes. In *Landscape Ecology Research Trends*; Dupont, A., Jacobs, H., Eds.; Nova Science Pub Inc.: Hauppauge, NY, USA, 2008; pp. 125–136.
12. Mitchell, N.; Rössler, M.; Tricaud, J.M. *World Heritage Cultural Landscapes—A Handbook for Conservation and Management*; UNESCO World Heritage Centre: Paris, France, 2009.
13. Folke, C. The economic perspective: Conservation against development versus conservation for development. *Conserv. Biol.* **2006**, *20*, 686–688. [\[CrossRef\]](#)
14. Martín-López, B.; García-Llorente, M.; Palomo, I.; Montes, C. The conservation against development paradigm in protected areas: Valuation of ecosystem services in the Doñana social-ecological system southwestern Spain. *Ecol. Econ.* **2011**, *70*, 1481–1491. [\[CrossRef\]](#)
15. Suzuki, N.; Parker, K.L. Potential conflict between future development of natural resources and high-value wildlife habitats in boreal landscapes. *Biodivers. Conserv.* **2016**, *25*, 3043–3073. [\[CrossRef\]](#)
16. Schmitz, M.F.; Arnaiz-Schmitz, C.; Herrero-Jáuregui, C.; Díaz, P.; Matos, D.G.; Pineda, F.D. People and nature in the Fuerteventura Biosphere Reserve (Canary Islands): Socio-ecological relationships under climate change. *Environ. Conserv.* **2017**, *45*, 20–29. [\[CrossRef\]](#)
17. *Ecosystems and Human Well-Being: Synthesis*; Island Press: Washington, DC, USA, 2005.
18. Karp, D.S.; Mendenhall, C.D.; Callaway, E.; Frishkoff, L.O.; Kareiva, P.M.; Ehrlich, P.R.; Daily, G.C. Confronting and resolving competing values behind conservation objectives. *Proc. Natl. Acad. Sci. USA* **2015**, *112*, 11132–11137. [\[CrossRef\]](#) [\[PubMed\]](#)
19. Costanza, R.; d’Arge, R.; de Groot, R.S.; Farber, S.; Grasso, M.; Hannon, B.; Limburg, K.; Naeem, S.; O’Neill, R.V.; Paruelo, J.; et al. The value of the world’s ecosystem service and natural capital. *Nature* **1997**, *387*, 253–260. [\[CrossRef\]](#)
20. Balmford, A.; Bruner, A.; Cooper, P.; Costanza, R.; Farber, S.; Green, R.E.; Jenkins, M.; Jefferiss, P.; Jessamy, V.; Madden, J.; et al. Economic Reasons for Conserving Wild Nature. *Science* **2002**, *297*, 950–953. [\[CrossRef\]](#) [\[PubMed\]](#)
21. Hattam, C.; Atkins, J.P.; Beaumont, N.; Börger, T.; Böhnke-Henrichs, A.; Burdon, D.; De Groot, R.; Hoefnagel, E.; Nunes, P.A.; Piwowarczyk, J.; et al. Marine ecosystem services: Linking indicators to their classification. *Ecol. Indic.* **2015**, *49*, 61–75. [\[CrossRef\]](#)
22. Geijzendorffer, I.R.; Martín-López, B.; Roche, P.K. Improving the identification of mismatches in ecosystem services assessments. *Ecol. Indic.* **2015**, *52*, 320–331. [\[CrossRef\]](#)
23. Sitas, N.; Prozesky, H.E.; Esler, K.J.; Reyers, B. Exploring the Gap between Ecosystem Service Research and Management in Development Planning. *Sustainability* **2014**, *6*, 3802–3824. [\[CrossRef\]](#)
24. Magliocca, N.R.; Rudel, T.K.; Verburg, P.H.; McConnell, W.J.; Mertz, O.; Gerstner, K.; Heinemann, A.; Ellis, E.C. Synthesis in land change science: Methodological patterns, challenges, and guidelines. *Reg. Environ. Chang.* **2015**, *15*, 211–226. [\[CrossRef\]](#)
25. Wang, Z.; Mao, D.; Li, L.; Jia, M.; Dong, Z.; Miao, Z.; Ren, C.; Song, C. Quantifying changes in multiple ecosystem services during 1992–2012 in the Sanjiang Plain of China. *Sci. Total Environ.* **2015**, *514*, 119–130. [\[CrossRef\]](#)
26. Metzger, M.J.; Rounsevell, M.D.A.; Acosta-Michlik, L.; Leemans, R.; Schröter, D. The vulnerability of ecosystem services to land use change. *Agric. Ecosyst. Environ.* **2006**, *114*, 69–85. [\[CrossRef\]](#)
27. TEEB. *The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations*; Pushpam, K., Ed.; Earthscan: London, UK; Washington, DC, USA, 2010.

28. Chaudhary, S.; McGregor, A.; Houston, D.; Chettri, N. The evolution of ecosystem services: A time series and discourse-centered analysis. *Environ. Sci. Policy* **2015**, *54*, 25–34. [[CrossRef](#)]
29. Quintas-Soriano, C.; Martín-López, B.; Santos-Martín, F.; Loureiro, M.; Montes, C.; Benayas, J.; García-Llorente, M. Ecosystem services values in Spain: A meta-analysis. *Environ. Sci. Policy* **2016**, *55*, 186–195. [[CrossRef](#)]
30. Hoyos, D.; Mariel, P.; Pacual, U.; Etxano, I. Valuing a Natura 2000 network site to inform land use options using a discrete choice experiment: An illustration from the Basque Country. *J. For. Econ.* **2012**, *18*, 329–344. [[CrossRef](#)]
31. Jobstvagt, N.; Hanley, N.; Hynes, S.; Kenter, J.; Witte, U. Twenty thousand sterling under the sea: Estimating the value of protecting deep-sea biodiversity. *Ecol. Econ.* **2014**, *97*, 10–19. [[CrossRef](#)]
32. Verma, M.; Negandhi, D.; Khanna, C.; Edgaonkar, A.; David, A.; Kadekodi, G.; Costanza, R.; Gopal, R.; Bonal, B.S.; Yadav, S.P.; et al. Making the hidden visible: Economic valuation of tiger reserves in India. *Ecosyst. Serv.* **2017**, *26*, 236–244. [[CrossRef](#)]
33. Zambrano-Monserrate, M.A.; Silva-Zambrano, C.A.; Ruano, M.A. The economic value of natural protected areas in Ecuador: A case of Villamil Beach National Recreation Area. *Ocean Coast. Manag.* **2018**, *157*, 193–202. [[CrossRef](#)]
34. Hicks, C.C.; McClanahan, T.R.; Hills, J.M.; Cinner, J.E. Trade-Offs in Values Assigned to Ecological Goods and Services Associated with Different Coral Reef Management Strategies. *Ecol. Soc.* **2009**, *14*, 10. [[CrossRef](#)]
35. Costanza, R.; De Groot, R.; Sutton, P.; Van Der Ploeg, S.; Anderson, S.J.; Kubiszewski, I.; Farber, S.; Turner, R.K. Changes in the global value of ecosystem services. *Environ. Chang.* **2014**, *26*, 152–158. [[CrossRef](#)]
36. Fisher, B.; Turner, K.; Zylstra, M.; Brouwer, R.; De Groot, R.; Farber, S.; Ferraro, P.; Green, R.; Hadley, D.; Harlow, J.; et al. Ecosystem Services and Economic Theory: Integration for Policy-Relevant Research. *Ecol. Appl.* **2008**, *18*, 2050–2067. [[CrossRef](#)]
37. Posner, S.; Getz, C.; Ricketts, T. Evaluating the impact of ecosystem service assessments on decision-makers. *Environ. Sci. Policy* **2016**, *64*, 30–37. [[CrossRef](#)]
38. Gómez-Baggethun, E.; de Groot, R. Natural capital and ecosystem services: The ecological foundation of human society. In *Ecosystem Services: Issues in Environmental Science and Technology*; Hester, R.E., Harrison, R.M., Eds.; Royal Society of Chemistry: London, UK, 2010; pp. 118–145.
39. IPBES. Preliminary Guide Regarding Diverse Conceptualization of Multiple Values of Nature and Its Benefits, Including Biodiversity and Ecosystem Functions and Services (Deliverable 3(d)); IPBES/4/INF/13; Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services: Bonn, Germany, 2015.
40. OpenNESS Project. Operationalisation of Natural Capital and Ecosystem Services. Available online: <http://www.openness-project.eu/> (accessed on 21 May 2019).
41. Jacobs, S.; Dendoncker, N.; Martín-López, B.; Barton, D.N.; Gomez-Baggethun, E.; Boeraeve, F.; McGrath, F.L.; Vierikko, K.; Geneletti, D.; Sevecke, K.J.; et al. A new valuation school: Integrating diverse values of nature in resource and land use decisions. *Ecosyst. Serv.* **2016**, *22*, 213–220. [[CrossRef](#)]
42. Castro, A.J.; Verburg, P.H.; Martín-López, B.; García-Llorente, M.; Cabello, J.; Vaughn, C.C.; Lopez, E. Ecosystem service trade-offs from supply to social demand: A landscape-scale spatial analysis. *Landsc. Policy Plan.* **2014**, *132*, 102–110. [[CrossRef](#)]
43. Bark, R.H.; Colloff, M.J.; Macdonald, D.H.; Pollino, C.A.; Jackson, S.; Crossman, N.D. Integrated valuation of ecosystem services obtained from restoring water to the environment in a major regulated river basin. *Ecosyst. Serv.* **2016**, *22*, 381–391. [[CrossRef](#)]
44. Liqueste, C.; Udias, A.; Conte, G.; Grizzetti, B.; Masi, F. Integrated valuation of a nature-based solution for water pollution control. Highlighting hidden benefits. *Ecosyst. Serv.* **2016**, *22*, 392–401. [[CrossRef](#)]
45. Villegas-Palacio, C.; Berrouet, L.; López, C.; Ruiz, A.; Upegui, A. Lessons from the integrated valuation of ecosystem services in a developing country: Three case studies on ecological, socio-cultural and economic valuation. *Ecosyst. Serv.* **2016**, *22*, 297–308. [[CrossRef](#)]
46. UNESCO. 2018. Available online: www.unesco.org/ (accessed on 15 December 2018).
47. Atauri, J.A.; de Pablo, C.; de Agar, P.; Schmitz, M.F.; Pineda, F.D. Effects of management on understory diversity in the forest ecosystems of northern Spain. *Environ. Manag.* **2004**, *34*, 819–828. [[CrossRef](#)]
48. Rescia, A.J.; Willaarts, B.A.; Schmitz, M.F.; Aguilera, P.A. Changes in land uses and management in two Nature Reserves in Spain: Evaluating the social-ecological resilience of cultural landscapes. *Landsc. Urban Plan.* **2010**, *98*, 26–35. [[CrossRef](#)]

49. Basque Government. *Governance Plan for Use and Management of Urdaibai Biosphere Reserve—Refunded Text*; Department of Land Management and Environment, Basque Government: Basque, Spain, 2004. (In Spanish)
50. Mateos, E.; Edeso, J.M.; Ormaetxea, L. Soil erosion and forests biomass as energy resource in the basin of the oka river in biscay, Northern Spain. *Forests* **2017**, *8*, 258. [CrossRef]
51. Onaindia, M.; Fernández de Manuel, B.; Madariaga, I.; Rodríguez-Loinaz, G. Co-benefits and trade-offs between biodiversity, carbon storage and water flow regulation. *Forest Ecol. Manag.* **2013**, *289*, 1–9. [CrossRef]
52. Basque Government. *geoEuskadi*. 2009. Available online: <http://www.geoeuskadi.eus/> (accessed on 15 September 2018).
53. European Union Habitats Directive (EUHD). *Council Directive 92/43/EEC, on the Conservation of Natural Habitats and Wild Flora and Fauna*; EUHD: Brussels, Belgium, 1992.
54. ESRI. *ArcGIS 10.4. Redlands*; Environmental Systems Research Institute: Redlands, CA, USA, 2016.
55. Burkhard, B.; Kroll, F.; Müller, F.; Windhorst, W. Landscapes' capacities to provide ecosystem services—A concept for land-cover based assessments. *Landsc. Online* **2009**, *15*, 1–22. [CrossRef]
56. Haines-Young, R.; Potschin, M.; Kienast, F. Indicators of ecosystem service potential at European scales: Mapping marginal changes and trade-offs. *Ecol. Indic.* **2012**, *21*, 39–53. [CrossRef]
57. Maes, J.; Egoh, B.; Willemen, L.; Liqueste, C.; Vihervaara, P.; Schägner, J.P.; Grizzetti, B.; Drakou, E.G.; La Notte, A.; Zulian, G.; et al. Mapping ecosystem services for policy support and decision making in the European Union. *Ecosyst. Serv.* **2012**, *1*, 31–39. [CrossRef]
58. Eigenbrod, F.; Armsworth, P.R.; Anderson, B.J.; Heinemeyer, A.; Gillings, S.; Roy, D.B.; Thomas, C.D.; Gaston, K.J. The impact of proxy-based methods on mapping the distribution of ecosystem services. *J. Appl. Ecol.* **2010**, *47*, 377–385. [CrossRef]
59. Herrero-Jáuregui, C.; Arnaiz-Schmitz, C.; Herrera, L.; Smart, S.M.; Montes, C.; Pineda, F.D.; Schmitz, M.F. Aligning landscape structure with ecosystem services along an urban–rural gradient. Trade-offs and transitions towards cultural services. *Landsc. Ecol.* **2018**, 1–21. [CrossRef]
60. Liqueste, C.; Cid, N.; Lanzasova, D.; Grizzetti, B.; Reynaud, A. Perspectives on the link between ecosystem services and biodiversity: The assessment of the nursery function. *Ecol. Indic.* **2016**, *63*, 249–257. [CrossRef]
61. Basque Government Statistics. Department of Economic Development and Infrastructures. 2017. Available online: <http://www.euskadi.eus/gobierno-vasco/estadisticas-agricultura-pesca-politica-alimentaria/inicio/> (accessed on 15 September 2018).
62. UBEI. Information of the State of the Water Bodies of the Basque Country. 2018. Available online: <http://www.uragentzia.euskadi.eus> (accessed on 15 December 2018).
63. Ruiz, J.M. *Modelo Distribuido para la Evaluación de Recursos Hídricos*; Monografías CEDEX M67; Ministerio de Fomento: Madrid, Spain, 1999.
64. MAPAMA. National Spatial Data Infrastructure. NDSI. 2015. Available online: <http://www.magrama.gob.es/es/cartografia-y-sig/ide/descargas/agua/simpa.aspx> (accessed on 15 September 2018).
65. Sharp, R.; Tallis, H.T.; Ricketts, T.; Guerry, A.D.; Wood, S.A.; Chaplin-Kramer, R.; Nelson, E.; Ennaanay, D.; Wolny, S.; Olwero, N.; et al. *INVEST 2.6.0 User's Guide*; The Natural Capital Project; World Wildlife Fund: Gland, Switzerland, 2018.
66. Neiker-Ihobe. *Estudio Sobre la Potencialidad de los Suelos y la Biomasa de Zonas Agrícolas, Pascícolas y Forestales de la CAPV Como Sumideros de Carbono*; Informe: Augusta, ME, USA, 2004.
67. IPCC. *Good Practice Guidance for Land Use, Land-Use Change and Forestry*; Institute for Global Environmental Strategies: Kanagawa, Japan, 2003.
68. Basque Government. *Forest Inventory 2011*; Department of Economic Development and Competitiveness: Basque, Spain, 2011.
69. Montero, G.; Ruiz-Peinado, R.; Muñoz, M. Monografías INIA: Serie Tierras forestales (13). In *Producción de Biomasa y Fijación de CO₂ por los Bosques Españoles*; Instituto Nacional de Investigación y Tecnología Agraria y Alimentaria (INIA) y Ministerio de Educación y Ciencia: Madrid, Spain, 2005.
70. CPF. Centre de la Propietat Forestal. Annex Indicadors dendromètrics. In *Manual de Redacció de Plans Tècnics de Gestió i Millora Forestal (PTGMF) i Plans Simples de Gestió Forestal*; PSGF: Barcelona, Spain, 2004; pp. 211–314.
71. Madrigal, A.; Álvarez, J.G.; Rodríguez, R.; Rojo, A. *Tablas de Producción para los Montes Españoles*; Fundación Conde del Valle de Salazar: Madrid, Spain, 1999.
72. Maes, J. *Water Retention Index*; European Commission, Joint Research Centre (JRC): Brussels, Belgium, 2010.

73. Peña, L.; Casado-Arzuaga, I.; Onaindia, M. Mapping recreation supply and demand using an ecological and a social evaluation approach. *Ecosyst. Serv.* **2015**, *13*, 108–118. [CrossRef]
74. Mamat, A.; Halik, Ü.; Rouzi, A. Variations of Ecosystem Service Value in Response to Land-Use Change in the Kashgar Region, Northwest China. *Sustainability* **2018**, *10*, 200. [CrossRef]
75. Brookshire, D.S.; Neill, H.R. Benefit transfers: Conceptual and empirical issues. *Water Resour. Res.* **1992**, *28*, 651–655. [CrossRef]
76. Richardson, L.; Loomis, J.; Kroeger, T.; Casey, F. The role of benefit transfer in ecosystem service valuation. *Ecol. Econ.* **2015**, *115*, 51–58. [CrossRef]
77. Brouwer, R. Environmental value transfer: State of the art and future prospects. *Ecol. Econ.* **2000**, *32*, 137–152. [CrossRef]
78. Johnston, R.J.; Rosenberger, R.S. Methods, trends and controversies in contemporary benefit transfer. *J. Econ. Surv.* **2010**, *24*, 479–510. [CrossRef]
79. Van Nes, E.H.; Scheffer, M. A strategy to improve the contribution of complex simulation models to ecological theory. *Ecol. Model.* **2005**, *185*, 153–164. [CrossRef]
80. Troy, A.; Wilson, M. Mapping ecosystem services: Practical challenges and opportunities in linking GIS and value transfer. *Ecol. Econ.* **2006**, *60*, 435–449. [CrossRef]
81. Bauer, D.M.; Johnston, R.J. The Economics of Rural and Agricultural Ecosystem Services: Purism versus Practicality. *Agric. Econ.* **2013**, *42*, iii–xv. [CrossRef]
82. Esteban Moratilla, F. *Proyecto VANE. Valoración de los Activos Naturales de España*; Ministerio de Medio Ambiente y Medio Rural y Marino, Gobierno de España: Madrid, Spain, 2010.
83. European Energy Exchange (EEX). EU Emission Allowances Secondary Market. 2018. Available online: <http://www.eex.com/> (accessed on 15 December 2018).
84. Busch, M.; La Notte, A.; Laporte, V.; Erhard, M. Potentials of quantitative and qualitative approaches to assessing ecosystem services. *Ecol. Ind.* **2012**, *21*, 89–103. [CrossRef]
85. Tol, R.S.J. The economic impact of climate change. *J. Econ. Persp.* **2009**, *23*, 29–51. [CrossRef]
86. Wang, P.; Deng, X.; Zhou, H.; Yu, S. Estimates of the social cost of carbon: A review based on meta-analysis. *J. Clean. Prod.* **2019**, *209*, 1494–1507. [CrossRef]
87. Zechter, R.; Kerr, T.M.; Kossoy, A.; Peszko, G. *Carbon Pricing Watch 2016*; World Bank Group: Washington, DC, USA, 2016.
88. Castillo-Eguskita, N.; Martín-López, B.; Onaindia, M. A comprehensive assessment of ecosystem services: Integrating supply, demand and interest in the Urdaibai Biosphere Reserve. *Ecol. Indic.* **2018**, *93*, 1176–1189. [CrossRef]
89. Van der Ploeg, S.; de Groot, R.S. *The TEEB Valuation Database—A Searchable Database of 1310 Estimates of Monetary Values of Ecosystem Services*; Foundation for Sustainable Development: Wageningen, The Netherlands, 2010.
90. Zhao, B.; Kreuter, U.; Li, B.; Ma, Z.; Chen, J.; Nakagoshi, N. An ecosystem service value assessment of land-use change on Chongming Island, China. *Land Use Policy* **2004**, *21*, 139–148. [CrossRef]
91. Mendoza-González, G.; Martínez, M.L.; Lithgow, D.; Pérez-Maqueo, O.; Simonin, P. Land use change and its effects on the value of ecosystem services along the coast of the Gulf of Mexico. *Ecol. Econ.* **2012**, *82*, 23–32. [CrossRef]
92. Gómez-Baggethun, E.; Barton, D.N. Classifying and valuing ecosystem services for urban planning. *Ecol. Econ.* **2013**, *86*, 235–245. [CrossRef]
93. Haase, D.; Larondelle, N.; Andersson, E.; Artmann, M.; Borgström, S.; Breuste, J.; Gómez-Baggethun, E.; Gren, Å.; Hamstead, Z.; Hansen, R.; et al. A Quantitative Review of Urban Ecosystem Service Assessments: Concepts, Models, and Implementation. *AMBIO* **2014**, *43*, 413–433. [CrossRef]
94. Luederitz, C.; Brink, E.; Gralla, F.; Hermelingmeier, V. A review of urban ES: Six key challenges for future research. *Ecosyst. Serv.* **2015**, *14*, 98–112. [CrossRef]
95. World Bank. *World Development Indicators*; World Bank Publications: Washington, DC, USA, 2009.
96. Mansfield, E. *Microeconomics: Theory and Applications*, 5th ed.; Norton: New York, NY, USA, 1985.
97. Kreuter, U.P.; Harris, H.G.; Matlock, M.D.; Lacey, R.E. Change in ecosystem service values in the San Antonio area, Texas. *Ecol. Econ.* **2001**, *39*, 333–346. [CrossRef]
98. Hao, F.; Lai, X.; Ouyang, W.; Xu, Y.; Wei, X.; Song, K. Effects of Land Use Changes on the Ecosystem Service Values of a Reclamation Farm in Northeast China. *Environ. Manag.* **2012**, *50*, 888–899. [CrossRef]

99. Aretano, R.; Petrosillo, I.; Zaccarelli, N.; Semeraro, T.; Zurlini, G. People perception of landscape change effects on ecosystem services in small Mediterranean islands: A combination of subjective and objective assessments. *Landsc. Urban Plan.* **2013**, *112*, 63–73. [\[CrossRef\]](#)
100. Kindu, M.; Schneider, T.; Teketay, D.; Knoke, T. Changes of ecosystem service values in response to land use/land cover dynamics in Munessa-Shashemene landscape of the Ethiopian highlands. *Sci. Total Environ.* **2016**, *547*, 137–147. [\[CrossRef\]](#)
101. Aschonitis, V.G.; Gaglio, M.; Castaldelli, G.; Fano, E.A. Criticism on elasticity-sensitivity coefficient for assessing the robustness and sensitivity of ecosystem services values. *Ecosyst. Serv.* **2016**, *20*, 66–68. [\[CrossRef\]](#)
102. Li, G.; Fang, C.; Wang, S. Exploring spatiotemporal changes in ecosystem-service values and hotspots in China. *Sci. Total Environ.* **2016**, *545–546*, 609–620. [\[CrossRef\]](#)
103. Song, W.; Deng, X. Land-use/land-cover change and ecosystem service provision in China. *Sci. Total Environ.* **2017**, *576*, 705–719. [\[CrossRef\]](#)
104. Schmitz, M.F.; De Aranzabal, I.; Pineda, F.D. Spatial analysis of visitor preferences in the outdoor recreational niche of Mediterranean cultural landscapes. *Environ. Conserv.* **2007**, *34*, 300–312. [\[CrossRef\]](#)
105. Burkhard, B.; Kandziora, M.; Hou, Y.; Müller, F. Ecosystem service potentials, flows and demand—Concepts for spatial localisation, indication and quantification. *Landsc. Online* **2014**, *34*, 1–32. [\[CrossRef\]](#)
106. Albert, C.; Bonn, A.; Burkhard, B.; Daube, S.; Dietrich, K.; Engels, B.; Frommer, J.; Götzl, M.; Grêt-Regamey, A.; Job-Hoben, B.; et al. Towards a national set of ecosystem service indicators: Insights from Germany. *Ecol. Indic.* **2016**, *61*, 38–48. [\[CrossRef\]](#)
107. Haines-Young, R.; Potschin, M.B. Common International Classification of Ecosystem Services (CICES) V5.1 and Guidance on the Application of the Revised Structure. 2018. Available online: www.cices.eu (accessed on 15 September 2018).
108. Duncker, P.S.; Raulund-Rasmussen, K.; Gundersen, P.; Katzensteiner, K.; De Jong, J.; Ravn, H.P.; Smith, M.; Eckmüllner, O.; Spiecker, H. How Forest Management affects Ecosystem Services, including Timber Production and Economic Return: Synergies and Trade-Offs. *Ecol. Soc.* **2012**, *17*, 50. [\[CrossRef\]](#)
109. Basque Government. Directorate of Biodiversity and Environmental Participation. In *Strategy of Sustainable Development of the Urdaibai Biosphere Reserve 2009–2015*; Technical Office of the Urdaibai Biosphere Reserve: Biscay, Spain, 2009.
110. Castillo-Eguskita, N.; Rescia, A.J.; Onaindia, M. Urdaibai Biosphere Reserve (Biscay, Spain): Conservation against development? *Sci. Total Environ.* **2017**, *592*, 124–133. [\[CrossRef\]](#)
111. *De 27 de Septiembre, por el que se Aprueba el Plan Rector de Uso y Gestión de la Reserva de la Biosfera de Urdaibai*; DECRETO 139/2016; Boletín Oficial del País Vasco: País Vasco, Spain, 2016.
112. Garmendia, E.; Mariel, P.; Tamayo, I.; Aizpuru, I.; Zabaleta, A. Assessing the effect of alternative land uses in the provision of water resources: Evidence and policy implications from southern Europe. *Land Use Policy* **2012**, *29*, 761–770. [\[CrossRef\]](#)
113. Rodríguez-Loinaz, G.; Amezaga, I.; Onaindia, M. Use of native species to improve carbon sequestration and contribute towards solving the environmental problems of the timberlands in Biscay, northern Spain. *J. Environ. Manag.* **2013**, *120*, 18–26. [\[CrossRef\]](#)
114. Naidoo, R.; Balmford, A.; Ferraro, P.; Polasky, S.; Ricketts, T.; Rouget, M. Integrating economic costs into conservation planning. *Trends Ecol. Evol.* **2006**, *21*, 681–687. [\[CrossRef\]](#)
115. Small, N.; Munday, M.; Durance, I. The challenge of valuing ecosystem services that have no material benefits. *Glob. Environ. Chang.* **2017**, *44*, 57–67. [\[CrossRef\]](#)
116. Gomez-Baggethun, E.; Martín-Lopez, B.; Barton, D.; Braat, L. *State-of-the-Art Report on Integrated Valuation of Ecosystem Services*; EU OpenNESS Project Deliverable 4.1; European Commission: Brussels, Belgium, 2014.
117. Martín-López, B.; Gómez-Baggethun, E.; García-Llorente, M.; Montes, C. Trade-offs across value-domains in ecosystem services assessment. *Ecol. Indic.* **2014**, *37*, 220–228. [\[CrossRef\]](#)
118. Farber, S.C.; Costanza, R.; Wilson, M.A. Economic and ecological concepts for valuing ecosystem services. *Ecol. Econ.* **2002**, *41*, 375–392. [\[CrossRef\]](#)
119. Gómez-Baggethun, E.; Ruiz-Pérez, M. Economic valuation and the commodification of ecosystem services. *Prog. Phys. Geog.* **2011**, *35*, 613–628. [\[CrossRef\]](#)

120. Schröter, M.; Van Der Zanden, E.H.; Van Oudenhoven, A.P.; Remme, R.P.; Serna-Chavez, H.M.; De Groot, R.S.; Opdam, P.; Serna-Chavez, H.M. Ecosystem Services as a Contested Concept: A Synthesis of Critique and Counter-Arguments. *Conserv. Lett.* **2014**, *7*, 514–523. [[CrossRef](#)]
121. Kallis, G.; Gómez-Baggethun, E.; Zografos, C. To value or not to value? That is not the question. *Ecol. Econ.* **2013**, *94*, 97–105. [[CrossRef](#)]
122. Adams, W.M. The value of valuing nature. *Science* **2014**, *346*, 549–555. [[CrossRef](#)]
123. Chan, K.M.A.; Balvanera, P.; Benessaiah, K.; Chapman, M.; Díaz, S.; Gómez-Baggethun, E.; Gould, R.; Hannahs, N.; Jax, K.; Klain, S.; et al. Opinion: Why protect nature? Rethinking values and the environment. *Proc. Natl. Acad. Sci. USA* **2016**, *113*, 1462–1465. [[CrossRef](#)]
124. Williams, B.K.; Brown, E.D. Adaptive Management: From More Talk to Real Action. *Environ. Manag.* **2014**, *53*, 465–479. [[CrossRef](#)]
125. Palacios-Agúndez, I.; Onaindia, I.; Barraqueta, P.; Madariaga, I. Provisioning ecosystem services supply and demand: The role of landscape management to reinforce supply and promote synergies with other ecosystem services. *Land Use Policy* **2015**, *47*, 145–155. [[CrossRef](#)]
126. Onaindia, M.; Peña, L.; De Manuel, B.F.; Rodríguez-Loinaz, G.; Madariaga, I.; Palacios-Agúndez, I.; Ametzaga-Arregi, I. Land use efficiency through analysis of agrological capacity and ecosystem services in an industrialized region (Biscay, Spain). *Land Use Policy* **2018**, *78*, 650–661. [[CrossRef](#)]
127. Li, B.-L. Why is the holistic approach becoming so important in landscape ecology? *Landsc. Urban Plan.* **2000**, *50*, 27–41. [[CrossRef](#)]



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